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Modelling incineration for more accurate comparisons to recycling in PEF and LCA

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ABSTRACT

When recycling is beneficial for the environment, results from a life cycle assessment (LCA) should give incentives to collection for recycling and also to the use of recycled material in new products. Many approaches for modeling recycling in LCA assign part of the environmental benefits of recycling to the product where the recycled material is used. For example, the Circular Footprint Formula in the framework for Product Environmental Footprints (PEF) assigns less than 45% of the benefits of recycling to a polymer product sent to recycling. Our calculations indicate that this creates an incorrect climate incentive for incineration of renewable LDPE, when the recovered energy substitutes energy sources with 100–300% more climate impact than the Swedish average district heat and electricity.

The risk of incorrect incentives can be reduced through allocating part of the net benefits of energy recovery to the life cycle where the energy is used; we propose this part can be 60% for Sweden, but probably less in countries without a district-heating network. Alternatively, the LCA can include the alternative treatment of waste that is displaced at the incinerator by waste from the investigated product.

These solutions both make the LCA more balanced and consistent. The allocation factor 0.6 at incineration almost eliminates the risk of incorrect incentives in a PEF of renewable polymers. However, the focus of LCA on one product at a time might still make it insufficient to guide recycling, which requires concerted actions between actors in different life cycles.

1. Introduction

1.1. Background

Recycling reduces the need for waste disposal and virgin material production. It brings a net environmental benefit when the environmental burdens of the avoided disposal and virgin production is greater than the burdens of recycling ($E^*_V + E^*_D - E_R > 0$, with the notation from Fig. 1). Waste incineration with energy recovery can also bring a net environmental benefit. Referring to Fig. 2, this happens when $E^*_E + E^*_D - E_{ER} > 0$. When the benefit of recycling is greater than the benefit of energy recovery ($E^*_V - E_R > E^*_E - E_{ER}$), results from a life cycle assessment (LCA) should guide decision-makers towards recycling: the results should indicate that collection for recycling is better than energy recovery; they should also indicate that the use of recycled material in new products is better for the environment, compared to the use of virgin material.

Most or all methods for modelling recycling in LCA risk giving incorrect incentives (Ekvall et al. 2020). Many approaches divide the benefits of recycling between the product supplying material to recycling and the product where the recycled material is used. This means the incentive for sending material to recycling is reduced. As an example, the Circular Footprint Formula (CFF) in the EU framework for Product Environmental Footprints (PEF) assigns the following burdens and benefits to a product that is recycled after use (E_{WR}; EC 2018a):

$$E_{WR} = (1 - A) \times \left(E_R - E^*_V \times \frac{Q_S}{Q_P}\right) \tag{1}$$

where A is a material-dependent factor that allocates part of the burdens

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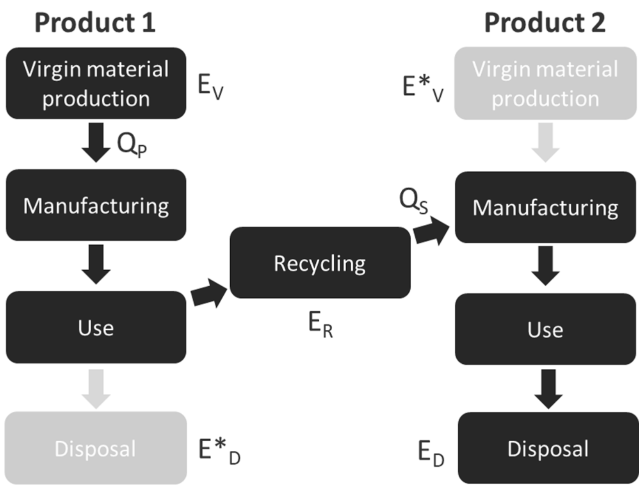

Fig. 1. Illustration of recycling. E_V, E_R and E_D are the environmental burdens of virgin material production, recycling, and final disposal, respectively. E^*_D and E^*_V are the environmental burdens of the disposal and virgin production avoided through recycling. Q_P and Q_S are the quality of primary and secondary material, respectively.

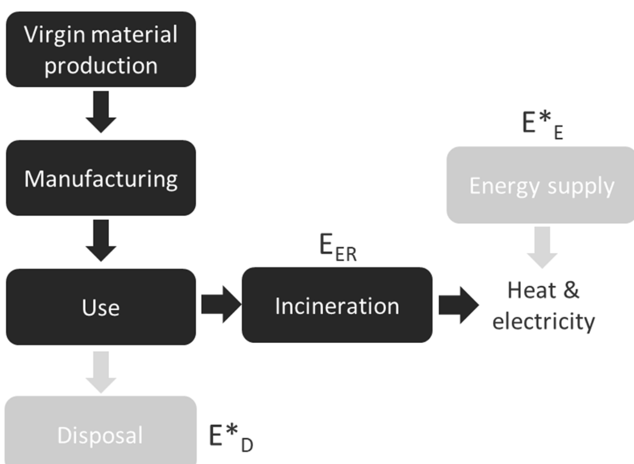

Fig. 2. Illustration of waste incineration with energy recovery from Product 1. E_{ER} is the environmental burdens of the incineration. E^*_E is the environmental burdens of the energy supply substituted by electricity and heat from the incineration.

and benefits of recycling to the product where the recycled material is used. Factor A is in the range of 0.2–0.8 and aims to reflect the market: A = 0.2 means the recycling is mainly limited by the supply of recycled material, while A = 0.8 means the recycling is mainly limited by the demand for recycled material. For most polymers, the default value of A is 0.5 and the default quality ratio Q_S/Q_P is 0.9 (EC 2018b). This means less than 45% of the net environmental benefit of recycling is assigned to the product being recycled (see Fig. 3).

The CFF assigns the burdens and benefits to a product that is incinerated after use (E_{WI}) in a similar manner:

$$E_{WI} = (1 - B) \times (E_{ER} - E_E^*) \tag{2}$$

Here, Factor B can be used to allocate part of the burdens and benefits of energy recovery to the life cycle where this energy is used. In contrast to Factor A, however, the default value Factor B is zero (EC 2018a). This means that CFF assigns the full net benefit of energy recovery to incinerated products.

In our hypothetical case of a polymer waste (Fig. 3), default PEF

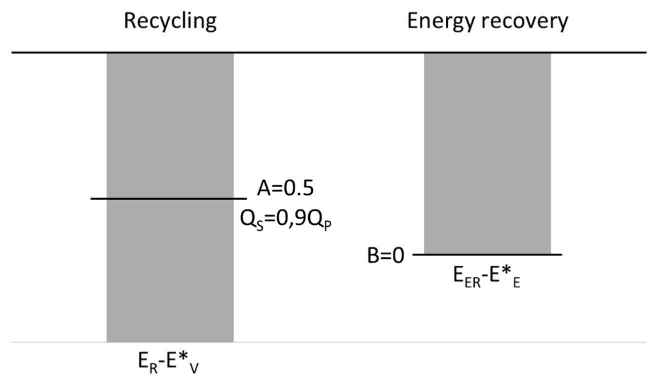

Fig. 3. The net environmental benefits of recycling and energy recovery in a hypothetical case of managing post-consumer polymer waste. Default PEF calculations (where Factor A = 0.5, secondary material has 90% of primary material quality, and Factor B = 0) in this case indicate that the waste should be incinerated even though recycling is better for the environment.

calculations assign more environmental benefit to the product if it is incinerated with energy recovery. The PEF results will indicate that the polymer waste should be incinerated, even though recycling is better for the environment (using the notation in Figs. 1-3: E^*_V-E_R > E^*_E-E_{ER}). When PEF is used for comparing polymer products, they might incorrectly indicate that a product designed for recycling is worse for the environment than a competing product. Similar problems can occur for products produced from other combustible materials. When PEF is used for product development, this removes the incentive for design for recycling. If PEF is used for policymaking, it can guide waste policy in the wrong direction.

The risk of incorrect incentives in the waste management disappears if the full environmental benefit of recycling is assigned to the product when recycled. This is the case with the end-of-life approach to modelling recycling. This rather common approach is also known as the closed-loop approximation, 0/100, recyclability substitution and value of scrap approach. It is recommended, for at least some cases, in several standards (ISO 2012; 2018a; 2018b; BSI 2011) and guidelines (JRC, 2010; Worldsteel, 2017; WRI & WBCSD, 2011; Fuji et al., 2005). However, an LCA with this approach assigns the same burdens to a product regardless of whether it is produced from recycled or virgin material. In other words, this approach gives no incentive to use recycled material.

A conceivable solution could be to assign the full benefit of recycling to the recycled product and also to the product using the recycled material. This approach to modelling recycling will always give a correct incentive to recycle and in addition give an incentive to use recycled material. However, the benefit of recycling is accounted for twice. A product produced from recycled material and recycled after use will be fully credited for the recycling at both ends of the life cycle. The double counting of the recycling benefit might make the total LCA results negative for such products. Such results give an incorrect incentive to produce the product even when it is not needed or wanted.

The problem of double counting can be alleviated if the recycling is modelled as a closed loop to the extent that the inflow and outflow of recycled materials match each other. A net inflow or a net outflow of recycled material would still be assigned the full benefit of recycling. However, this solution would completely remove the incentive to recycle a product that is produced from 100% recycled materials: such a product would be assigned the full benefit of recycling whether it is recycled after use or not.

1.2. Purpose and method

This paper presents a pilot project (funded by the Swedish Environmental protection Agency) that aims to investigate whether there is a real risk that results from PEF and other LCAs give incorrect incentives for incineration in Sweden. We also present and assess two possible solutions to amend this problem by a renewed analysis of the modelling of energy recovery: 1) use of Factor B in the PEF methodology, and 2) a broader systems perspective that includes the effects of energy recovery on waste imports and thus waste management in other countries. Although the focus is on Swedish conditions, we discuss the implications for other countries. The methods proposed should be applicable in all countries where energy is recovered through waste incineration.

We analyze the risk of incorrect incentives and the proposed solutions through a simple case study on waste management of low-density polyethylene (LDPE) produced from renewable raw materials. We estimate the net climate benefit of mechanical recycling (E^*_{Vm}-E_{Rm}), chemical recycling (E^*_{Vc}-E_{Rc}) and incineration (E^*_{E}-E_{ER}) through simple substitution. We apply the CFF with default values ($A = 0.5$; $Q_S/Q_P = 0.9$; $B = 0$) to find whether a PEF would give the same indications as a simple substitution. We then modify CFF by applying a revised Factor B in the CFF to investigate whether this brings PEF results more in line with the estimated net climate benefit of recycling and incineration with energy recovery. Finally, we modify the CFF with two scenarios on how a change in the incineration of LDPE waste would affect waste imports and, hence, the waste management in other European countries.

The results are used as basis for a discussion aiming to draw conclusions about the extent to which PEF, and LCA in general, risk giving incorrect incentives for energy recovery, and about the extent to which there is a good basis for methods to alleviate this problem. Further details are given in a project report (Ekvall et al. 2021), which presents an earlier version of this study.

2. Factor B

2.1. Interpreting Factor B

The CFF models both recycling and energy recovery with substitution (cf. Equations (1) and (2)): it includes the material and energy production substituted through recycling and energy recovery. Substitution at waste management is a widely established practice, particularly in consequential LCA (CLCA), i.e., LCAs aiming to estimate how the global environmental impacts are affected by a decision.

When applying substitution, it can be useful to distinguish between determining and dependent co-products. The production volume of the process is determined by the demand for the determining product, and not affected by the demand for dependent co-products. Instead, dependent co-products are produced in volumes decided by the demand for the determining co-product (see Fig. 4). A CLCA of a determining co-product will include the joint production process and a credit for the avoided supply of competing products substituted by dependent co-products. A CLCA of a dependent co-product will not include the joint production since it is not affected by the demand for the product. Instead, it will include the affected, alternative supply of the competing product. This means the use of the dependent co-product in an CLCA is assigned the burdens of the alternative supply.

Using the default value $B = 0$ in the energy substitution is equivalent to assuming that the waste treatment service is the determining function of energy recovery, i.e., that the volume of waste incinerated is solely determined by the quantity of combustible waste (cf. Equation (2) and Fig. 5). This assumption is often wrong. In many countries, combustible waste is deposited at landfills and the volume incinerated is much less than the volume of combustible waste. In Sweden, on the other hand, waste incinerators are constructed even though the existing capacity is more than enough to treat the domestic combustible waste that is not recycled (Waste Sweden, 2012a; Profu, 2013). Of the nearly 6 million tonnes of waste combusted in Swedish waste incinerators in 2018, 1.5 million tonnes were waste from other European countries (Waste Sweden, 2019).

The expansion of waste incineration in Sweden is driven by good economic conditions for incinerators in the country. The energy in the waste is used more efficiently in Sweden, compared to many other countries, because the heat can be used in district-heating networks. The Swedish taxes on fossil fuels also make waste a more competitive fuel (Bisaillon et al. 2005).

Waste incineration has higher investment costs than other fuels since waste incinerators require advanced technology for combustion as well as flue gas treatment. On the other hand, the variable cost of energy recovery from waste is very low, if not negative, because of the gate fee that incinerators charge for accepting the waste. Therefore, waste incineration plants are base-load plants that are typically used to their full capacity. When free incineration capacity occurs, waste is imported to fully utilize the incineration capacity. Hence, the volume of waste incinerated, and the corresponding energy recovery is mainly determined by the waste-incineration capacity.

2.2. Approach for calculating B

The concept of determining functions provides a basis for calculating a value of B that differs from zero. From the above we observe that 1) the quantity of energy recovered from waste is determined mainly by the waste incinerator capacity, and 2) increases in waste incinerator capacity are determined by the expected profitability. Heat, electricity, and gate fees paid to deliver the waste all contribute to this profitability (see Fig. 5). Hence, waste incineration can be described as a process with multiple determining functions: waste treatment and energy recovery contribute to driving the process in proportion to their economic value.

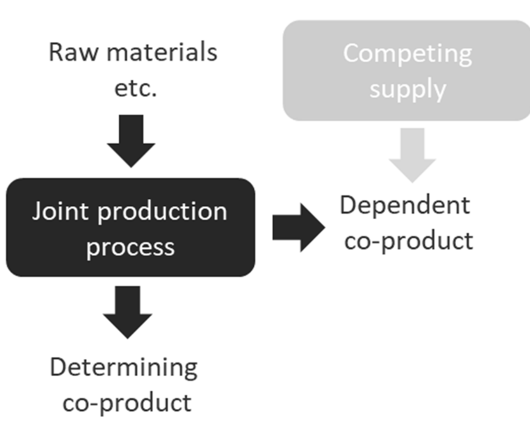

Fig. 4. Distinction between determining and dependent co-products, simplified from Weidema (2001). Demand for the determining co-product affects both the joint production and the competing supply of dependent co-products. Demand for a dependent co-product affects only the alternative supply of this product.

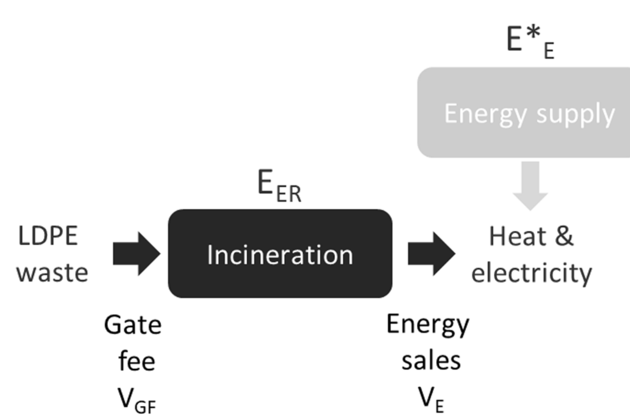

Fig. 5. Waste incineration has two functions: waste treatment and energy recovery. Is waste treatment the determining function? V_{GF} and V_E are the expected economic value of gate fees and energy, respectively.

We propose that expected revenues from gate fees and energy are an appropriate basis for calculating Factor B:

$$B = V_E / (V_E + V_{GF}) \quad (3)$$

where V_E and V_{GF} are the expected economic value of energy and gate fees, respectively.

2.3. Factor B in Sweden and elsewhere

The national trade organization Waste Sweden (2014) recommends economic allocation of emissions from waste incineration with 58.7% allocated to the energy and the remaining 41.3% to the waste treatment. Using the same values for calculating B we get B = 0.587. The economic revenues will vary with time and between locations. In Gothenburg, for example, the energy generates only 30% of the revenues in a waste-management system dominated by incineration (Renova, 2020). Factor B should ideally be calculated based on updated data on expected revenues in the relevant region. However, when such data are lacking, we propose that a default value B = 0.6, rounded from 0.587, can be used for Sweden.

In countries without district-heating networks, the energy revenues are likely to be lower. A higher share of the cost of incineration will then be covered by the gate fee. This means Factor B is likely to be lower than 0.6.

3. A wider systems perspective

3.1. Impacts in Sweden

The capacity of waste incinerators is often defined by the heat exchangers, which sets a limit for the energy content of the waste flow. When we send polymer waste to incineration, this means we do not necessarily get more energy from the incinerator. Instead, the polymer waste is likely to replace other waste flows with similar amount of energy.

Landfilling of combustible waste is not allowed in Sweden. Hence, an increase in incineration of waste polymers will not lead to an increase in landfilling of other waste flows in Sweden. Instead, it is primarily the import of combustible waste that is reduced. Several previous reports indicate that imported waste is the marginal fuel in Swedish waste incineration plants (Gode et al., 2013; Waste Sweden, 2012b; cf. Fig. 6).

3.2. Impacts in other countries

Sweden imports large amounts of combustible mixed waste (containing biological waste, polymers, textiles, inert materials, etc.) from Norway and the United Kingdom. Smaller quantities are imported from Ireland, the Netherlands, Finland, and other countries. Fråne et al. (2016) examined how this trade in waste affects waste management in exporting countries, but found that the causal relationships are often difficult to identify. Exports of waste from Norway increased sharply when a ban on landfilling of biological waste was introduced there in 2009. In at least one case, the possibility of exporting food waste as part of mixed residual waste to Sweden contributed to impeding investments in infrastructure for source separation of food waste in Norway. Waste exports to Sweden may also have affected the expansion of incineration plants in Norway and can affect exports of waste for incineration in other European countries (Fråne et al. 2016).

The waste exported from the UK is largely residues from sorting facilities (so-called Material Recovery Facilities; MRF) or from Mechanical Biological Treatment (MBT). Most of this is sent to the Netherlands for energy recovery, and only a small share is exported to Sweden. It is possible to deposit the waste in UK landfills, but this is expensive due to high landfill taxes. The option of sending residues for energy recovery in Sweden is cheaper. However, the Swedish gate fees are set just low enough to compete with other options for treating the residues. Hence, the export to Sweden probably has no significant impact on the economy of the MRF and MBT facilities and hardly affects the investments or operation of such facilities. Instead, a change in the export to Sweden primarily affects how much is deposited in the United Kingdom and how much is exported for incineration in other countries (Fråne et al. 2016).

In Europe as a whole, the incineration capacity is significantly smaller than the supply of combustible waste. Large amounts of combustible waste are still landfilled, even if an increasing share of the deposited waste first goes through MBT or other pre-treatment processes. If a reduction in Swedish waste imports means that the countries of origin instead export the waste to other countries for incineration, the limited capacity for incineration there, and in the rest of Europe, means that other waste will probably be replaced in the incinerators and landfilled somewhere in Europe.

In general, Fråne et al. (2016) argue that Swedish waste imports contribute to keeping costs down for waste treatment outside Sweden's borders. In theory, this makes it more difficult for material recycling to compete. However, since the Swedish facilities negotiate gate fees that are just below the cost of other waste treatment, the effect on recycling is likely to be very small in the foreseeable future.

The EU aims to reduce landfilling significantly, and the capacity for incineration and other waste treatment is therefore being expanded in Europe as a whole. This could in the long run lead to a European overcapacity for incineration. If so, there will be competition for the combustible waste, resulting in lower gate fees. In such a situation, a reduced import of waste to Sweden can contribute to keeping the gate fees up. This can lead to increased material recycling, for example through an increased degree of sorting in MRF facilities or through increased investments in infrastructure for source sorting. However, increased gate fees can also lead to increased investments in energy recovery and/or biological treatment, or to increased landfilling in the countries where this is permitted.

3.3. Our scenarios

As clear from above, a reduction in import of combustible waste to Sweden can have a range of impacts on the waste management in other countries. The actual consequences are likely to be a combination of, for example: 1) increased disposal of untreated residual waste in the exporting countries or elsewhere in Europe, 2) increased disposal of MBT and MRF residues in the countries of origin or elsewhere in Europe, 3) (at least in the long run) increased biological treatment in other countries, 4) (in the long run) increased incineration in other countries, and/or 5) (in the long run) increased material recycling.

The uncertainty is great in what effects will dominate the mix of consequences. This uncertainty can be illustrated and managed with scenarios. The scenarios should be widely separate without being

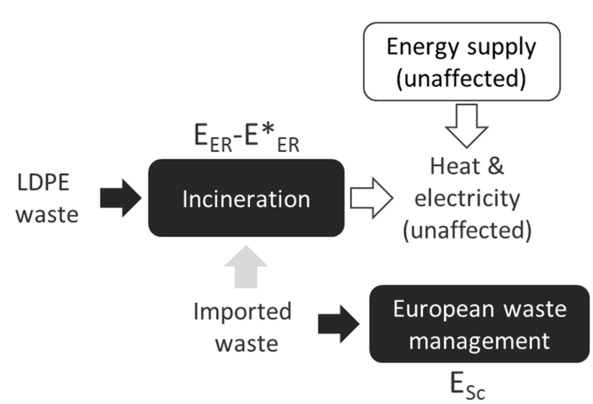

Fig. 6. A wider systems perspective: LDPE incineration affects waste imports, which in turn affects waste management elsewhere in Europe. E_{Sc} represents the net environmental burdens of the affected European waste treatment.

unreasonable. They should also be simple enough to communicate and understand. We have chosen two scenarios based on Hagberg et al. (2017): European incineration and European landfill.

If Swedish plastic waste is incinerated rather than recycled, imports of combustible mixed waste are reduced. In the scenario "European incineration", this results in more waste being incinerated with electricity production (33% efficiency) in another European country. The electricity substitutes electricity produced from natural gas in modern combined cycle power plants with 58% efficiency. The latter is a deviation from Hagberg et al. (2017), who assumed the substituted natural gas to be produced in an older, less efficient plant.

In "European landfill", increased incineration of plastic waste in Sweden leads to reduced waste imports, which in this scenario leads to an increase in the disposal of untreated residual waste in another European country. The combustible mixed waste is disposed in a modern, well-designed landfill, where 70% of the methane formed during a hundred-year period is utilized as landfill gas. The extracted landfill gas is assumed to be used with a total efficiency of 80% to produce electricity (25%) and heat (75%), which displace electricity and heat produced from natural gas. Of the unextracted methane, 10 % (i.e., 3% of the generated methane) is assumed to oxidize in the landfill cover and not affect the climate.

4. Calculation example

This section includes a simple case study aiming to investigate and illustrate the risk of LCA providing incorrect incentives for energy recovery from plastic waste, and to test and illustrate to what extent a modified Factor B or a broader systems perspective solves this problem.

4.1. Goal and scope definition

The goal of the case study is to illustrate and test methodological approaches for modelling recycling and energy recovery. The object of study is the climate impacts of mechanical recycling, chemical recycling through pyrolysis, and incineration with energy recovery of waste bio-based LDPE. The calculations are valid for renewable LDPE waste generated and treated in Sweden in the near future: 1–5 years from now.

We compare the three waste-management options with four different methodological approaches: 1) simple substitution, where the full benefit of recycled material substituting virgin material is assigned to the LDPE sent to recycling, and the full benefit of avoided heat and electricity production is assigned to the LDPE sent to incineration, 2) the default PEF approach, 3) PEF calculations with B = 0.6 (cf. Section 2.3), and 4) adjusted PEF calculations with two scenarios accounting for the impact in other European countries of a change in Swedish waste imports (cf. Section 3.3).

In all four approaches, the functional unit is the treatment of 1 tonne waste LDPE. The comparison between incineration and recycling is made possible through expanding the system to include avoided production of energy and virgin material. In the fourth approach, the system is further expanded to include not only the reduced incineration of imported waste, but also the induced landfilling or incineration in other countries, and the energy supply substituted by energy from this waste treatment.

Since the object of study is the waste treatment, the calculations do not include the production and use of the LDPE product. Emissions of biogenic CO_2 are considered climate neutral and, hence, excluded from the calculations.

4.2. Inventory analysis

We create models for mechanical recycling, chemical recycling and incineration with energy recovery in the GaBi Software. The scenarios with a European systems perspective are calculated with WAMPS (WAste Management Planning System), an LCA model for waste management (Moora et al. 2006), with key assumptions listed in Section 3.3. Waste incinerator data are from WAMPS. Most other input data are from Thinkstep/Sphera and EcoInvent databases available in GaBi – with a few exceptions mentioned below. To keep the study simple, we accept data gaps when we do not expect them to matter for the total results. This includes CH_4 emissions from waste incinerator plants.

We make the following assumptions: each tonne of mechanically recycled LDPE displaces 1 tonne of virgin LDPE granules produced in Europe from fossil raw materials; the 720 kg pyrolysis oil generated per tonne of LDPE in chemical recycling (Lassesson et al. 2021) displace the same amount of naphtha produced from fossil resources; and the heat and electricity generated at waste incineration displace the same amount of average Swedish heat and electricity. The impacts of these assumptions are discussed in the life cycle interpretation (Section 4.4).

The calculations account for international transports, but not for local and regional transports. We account for the climate impact of sorting, baling and preprocessing of LDPE going to recycling; however, this impact (51 kg CO_2-eq. per tonne recycled LDPE) is not important for the total results, and we assume it is the same for both recycling options.

In the PEF approach, recycling is modelled with Equation (1). For mechanical recycling, we use the PEF default value for polymers: A = 0.5 and $Q_S/Q_P = 0.9$. When assessing chemical recycling, we set $Q_S = Q_P$, because there is no significant difference in quality between the pyrolysis oil and the naphtha. Factor A is still 0.5, because this is the default value for materials lacking other default values (EC 2018b).

To account for the impacts of changes in the Swedish waste import on the waste management in other countries, we apply Equation (4), which is a modified version of Equation (2):

$$E_{WI} = (1 - B) \times (E_{ER} - E^{*}_{ER} + E_{Sc}) \tag{4}$$

where E^{*}_{ER} is the climate impact of displaced Swedish incineration of imported residual waste, and E_{Sc} is the scenario-dependent net climate impact of treating this waste in other European countries (cf. Section 3.3).

The lower heating value (LHV) of LDPE is 42.8 GJ/tonne (Phyllis, 2020). We assume the LDPE to replace a residual European waste mix that includes 11% fossil plastics and has a LHV of 10.6 MJ/kg, which is the default value in WAMPS. This means each tonne of LDPE displaces 4.04 tonne residual waste at the incinerator, because we assume the capacity of the waste incinerator to be limited by the energy content of the waste flow. The impact of this assumption is discussed in Section 4.4.

The results of the inventory analysis are summarized in Table 1, disaggregated into the emissions from LDPE waste treatment and the net impacts of emissions from the substituted systems.

4.3. Impact assessment

For the impact assessment we use characterization factors for the hierarchist perspective from the ReCiPe 2016 v1.1 (Huijbregts et al. 2017): 0 for biogenic CO_2, 1 for fossil CO_2, 34 for biogenic CH_4, 36 for fossil CH_4, and 298 for N_2O. These are based on numbers published by the IPCC (2013) with a 100-year time horizon and accounting for climate-carbon feedbacks.

Impact assessment results are presented in Table 1 and Fig. 7. They are dominated by the benefit (or impact) of substituting systems related to virgin material production, energy supply, and incineration of imported waste.

The results indicate a large net climate benefit of mechanical LDPE recycling: 1.8 tonne CO_2-eq. per tonne recycled LDPE waste. Chemical recycling of renewable LDPE brings a much smaller climate benefit, even though we assume emissions of bio-based CO_2 from the pyrolysis to be climate-neutral. This is partly because the climate benefit of substituting naphtha is relatively small: 163 kg CO_2-eq. per tonne LDPE.

The climate impact of incineration (E_{ER}) is low in our results (90 kg CO_2-eq. per tonne LDPE). Biogenic CO_2 is excluded from the calculation,

Table 1

Results from the calculations to illustrate and test methodological approaches for modelling recycling and energy recovery. The table includes greenhouse gas emissions and their climate impact associated with treatment of 1 tonne waste renewable LPPE. All numbers are rounded to a maximum of two significant digits. Negative numbers indicate a reduction in emissions, i.e., a climate benefit.

	kg CO_2 (fossil)	kg CH_4	kg N_2O	kg CO_2 equivalents
Mechanical recycling				
Recycling, incl. pretreatment (E_R)	48	0.0	0.003	51
Avoided LDPE production (E^*_V)	−1800	−3.1	−0.030	−1900
Net total (simple substitution)	*−1700*	*−3.1*	*−0.027*	*−1800*
Chemical recycling				
Recycling, incl. pretreatment (E_R)	48	0.0	0.003	51
Avoided naptha production (E^*_V)	−88	−2.1	−0.002	−160
Net total (simple substitution)	*−40*	*−2.0*	*0.001*	*−110*
Energy recovery				
Incineration (E_{ER})	12	0.1	0.25	90
Avoided heat production	−370	−0.4	−0.083	−410
Avoided electricity production	−46	0.0	−0.003	−48
Avoided energy supply (E^*_E)	−420	−0.4	−0.086	−460
Net total (simple substitution)	*−410*	*−0.3*	*0.16*	*−370*
Displaced incineration (E^*_{ER})	−2100	0.0	−0.25	−2200
Incineration in other countries (E_{SC})	970	−3.7	0.25	920
Net total, European incineration	*−1100*	*−3.6*	*0.25*	*−1200*
Landfilling in other countries (E_{SC})	−1200	150	1.08	4100
Net total, European landfilling	*−3300*	*150*	*1.1*	*2000*

and we have a data gap for direct CH_4 emissions from the incinerator, which means the E_{ER} is dominated by N_2O emissions and by CO_2 emissions from the production of fuel and electricity used at the incineration site. However, the climate benefit of energy recovery is also modest, due to the low climate impact of Swedish average district heat and electricity. In the three cases where waste incineration displaces average Swedish heat and electricity (Fig. 7a-c), the net climate benefit of incinerating renewable LDPE waste (0.37 tonne CO_2-eq. per tonne LDPE) is greater than for chemical recycling, but smaller compared to mechanical recycling.

When incineration is modelled with displaced imports of waste (Fig. 7d), the climate impact of the displaced Swedish incineration of imported residual waste (E^*_{ER} in Equation (3)) is 2.2 tonne CO_2-eq. per tonne LDPE. The climate impact of treating this waste in other European countries (E_{Sc}) is scenario-dependent: 0.9 tonne CO_2-eq. per tonne LDPE in “European incineration”, and 4.1 tonne CO_2-eq. per tonne LDPE in “European landfill”. In the scenario “European incineration”, Swedish incineration of renewable LDPE waste results in a significant net climate benefit by increasing electricity production in waste incinerators in other countries. In “European landfilling”, Swedish incineration of renewable LDPE waste results in a significant climate impact by increasing landfilling in other countries.

4.4. Life cycle interpretation

Some of our assumptions (cf. Section 4.2) have moderate impact on the results. If each tonne recycled LDPE displaces just 0.9 tonne virgin LDPE, the climate benefit of mechanical recycling is 10% lower than indicated in our results. If the 720 kg of pyrolysis oil substitutes something else than 720 kg naphtha, this will affect the relatively small climate benefit of chemical recycling.

Other assumptions are more important. The net total results for incineration are heavily affected by the assumption that heat and electricity from the incineration substitutes average Swedish heat and electricity. The benefit of energy recovery from renewable plastics would be significantly greater if the calculations accounted for marginal impacts in the district-heat and electricity systems. The net benefit of incineration would also be greater if the substituted electricity were modelled with data on Nordic residual electricity, as stipulated for a Swedish PEF (EI, 2021); however, this has a limited effect on our PEF results, since only a small share of the recovered energy is electricity.

When accounting for impacts on waste imports, the results are sensitive to the assumption that the capacity of the waste incinerator is limited by the energy flow. If the capacity is instead limited by the mass flow of the waste, each tonne of renewable LDPE displaces 1 tonne of mixed waste instead of 4 tonnes. The net benefit or impact of the European scenarios are then just 25% of what is indicated by Table 1. Furthermore, energy output from the incinerator increases because the LDPE waste contains four times more energy than the mixed waste. This means the LDPE incineration gets a credit for avoided Swedish heat and electricity production, independent of the scenario. With all other assumptions and input data the same, the total net results would be −0.6 tonne CO_2-eq. per tonne recycled LDPE waste with European incineration and −0.2 tonne CO_2-eq. with European landfilling, compared to −1.2 and +2.0 tonne CO_2-eq. in Table 1.

The results for the European scenarios also depend heavily on scenario-specific assumptions (cf. Section 3.3). The assumption regarding what electricity source is substituted is important in the scenario European incineration. In European landfilling, assumptions on the creation and capture of methane in landfills are particularly important.

The climate impact of LDPE incineration is highly uncertain due to the data gap on CH_4 emissions and uncertain N_2O data. However, the net total results are not heavily influenced by this uncertainty, since emissions from incineration of renewable LDPE are not very important for the net total results.

Limitations in our scope can be more important for the total results. When renewable plastics are recycled and substitute fossil plastics in new products, a climate benefit can arise at the end-of-life of these products: if they are incinerated after use, the CO_2 emissions from the incineration will be biogenic and, hence, climate neutral. This impact is not accounted for in our calculations, which means the climate benefit of recycling can be even greater than our results indicate.

5. Discussion and conclusions

5.1. The risk of incorrect incentive

As stated in Section 1.1, recycling is better for the environment when E^*_V-E_R > E^*_E-E_{ER} (cf. Figs. 1 and 2). Such results are obtained with simple substitution. Our results from simple substitution (Fig. 7a) indicate that the net climate benefit of energy recovery is much smaller than the benefit of mechanical recycling. However, the benefit of energy recovery is underestimated in our calculations (cf. Section 4.4). If the climate impact of the substituted energy were four times greater than the average Swedish district heat and electricity, the benefit of energy recovery and mechanical recycling would be approximately the same.

From this we conclude that recycling to replace virgin LDPE is better for the climate than energy recovery, when the climate impact of the substituted energy is not more than 300% greater than the climate impact of average Swedish energy. In these cases, LCA results should indicate that products should be sent to mechanical recycling and new

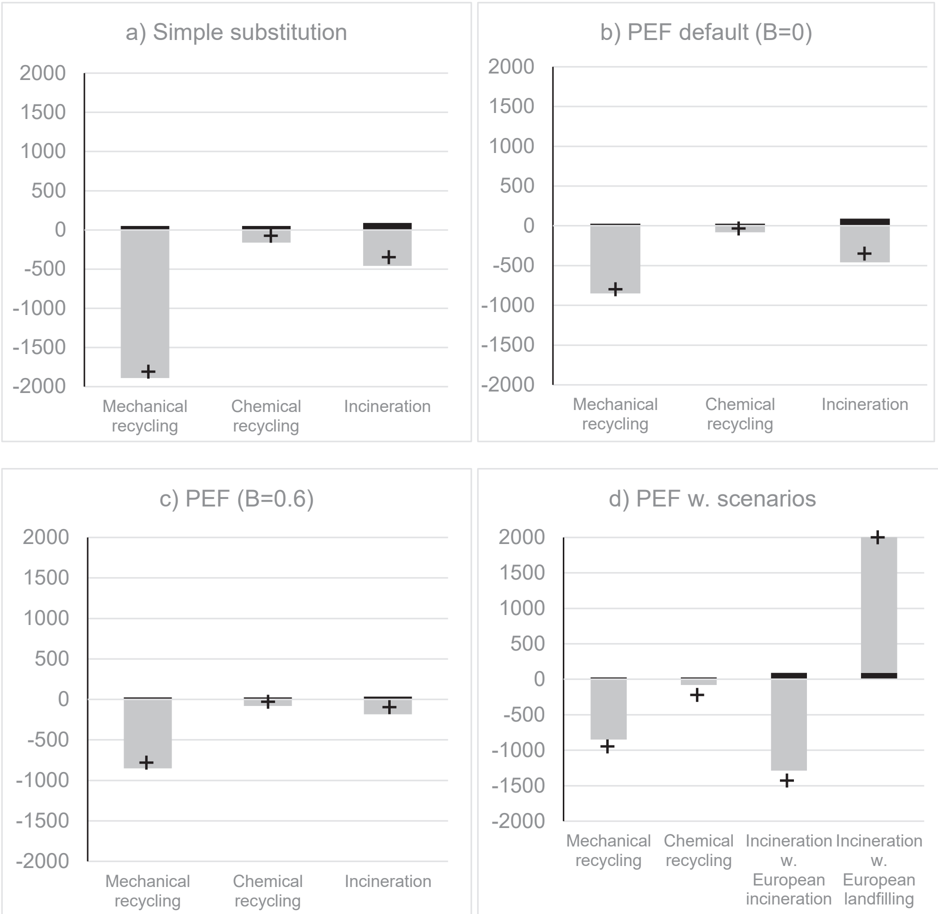

Fig. 7. Climate impacts (in kg CO_2 equivalents per tonne LDPE waste) of mechanical recycling, chemical recycling and incineration with energy recovery of renewable LDPE assessed with a) simple substitution as an estimate of actual benefits of recycling and energy recovery; b) CFF with default values; c) CFF with B estimated for Sweden (B = 0.6); and d) CFF modified with European scenarios to account for impacts of a change in import of combustible waste. Positive bars imply an increase in emissions, while negative bars indicate a reduction in emissions. Black parts of the bars represent the climate impact of the waste-treatment system. Grey parts represent the net benefit or impact of substituting other systems. The net total is marked with a plus sign.

products should be produced from recycled materials.

In the default PEF approach (Fig. 7b), less than 45% of the environmental net benefit of recycling is assigned to the LDPE products recycled after use. The results still indicate that mechanical recycling is a better option for the climate, compared to energy recovery. In other words, they give a correct incentive to send waste LDPE to mechanical recycling.

However, the difference between mechanical recycling and incineration is less than in Fig. 7a. If the substituted energy has more than double the climate impact of the average Swedish district heat and electricity, default PEF results would indicate that renewable LDPE should be incinerated. Hence, our results indicate that default PEF calculations will give an incorrect incentive to incinerate renewable LDPE waste, when the energy substituted by incineration has 100–300% more climate impact than the average Swedish energy.

Our calculations are made for Swedish conditions. However, PEF risks giving incorrect incentives for incineration and other energy-recovery processes everywhere when energy recovery from renewable LDPE brings a net environmental benefit that is 45–100% of the net environmental benefit of recycling (cf. Fig. 3). If the benefit of incineration is less than 45% of the benefit of recycling, default PEF calculations are likely to correctly indicate that the polymer should be recycled. If the benefit of incineration is greater than the benefit of recycling, PEF results will correctly indicate that energy recovery is the better option.

The energy content and climate impact of virgin production vary between polymers; however, the range of variation tends to be limited, with the exception of PVC. The default values for Factor A and Q_S/Q_P are also the same for most polymers (EC 2018b). For this reason, we expect the risk for incorrect incentives in PEF results to be about the same for other recyclable, bio-based polymers. If the polymer is produced from fossil resources, there is no such risk: PEF results will not indicate that the waste polymer should be incinerated, because incineration of fossil polymers is associated with a large climate impact.

Default PEF calculations risk giving incorrect incentives for incineration also for other bio-based waste materials. For most paper grades the risk is relatively small, because 80% of the net environmental benefit of recycling is assigned to paper products recycled after use. For textiles, the risk might be much greater, because only 20% of the net benefit of recycling is assigned to textile products recycled after use.

Similar problems occur with other methods that account for the fact that supply and demand for recyclable material are both needed for recycling to occur: for example, price-based allocation (ISO 2018a), 50/50 allocation (Lindfors et al. 1995), and 50/50 substitution (Ekvall 2000). Even ambitious methods that aim to model the foreseeable consequences of sending a material to recycling can fail to give correct information on when a material should be sent to recycling: for example, the price elasticity approach (Ekvall 2000), and price-based substitution (Schrijvers et al. 2016).

Accurate modelling the consequences of sending a material to recycling is clearly not enough to always give correct indication regarding how to treat the waste. This might be because recycling requires a concerted action between actors in different product life cycles, and LCA focusses on one product at a time. When actions are needed in different life cycles, the focus on individual life cycles appears to be an important limitation with LCA.

To clarify with a drastic example: consider assessing the foreseeable consequences of clapping your right hand. The hand is most likely to hit air only. It might also hit an object or person in front of you. The chances that you are going to hurt yourself or someone else is much greater than the likelihood that your right hand is going to produce a clapping sound.

In conclusion, you should refrain from clapping your right hand. The same goes for the left hand, of course. Hence, there will be no applaud.

5.2. Factor B as a solution

When B is changed from 0 to 0.6, the environmental burdens and benefits of incineration are reduced by 60%, compared to the default PEF results (see Equation (2)). Compared to simple substitution (Fig. 7a), our PEF calculations with B = 0.6 (Fig. 7c) reduce the net results of recycling by just over 55% and the net results of incineration by 60%. This means the results will point in the same direction as simple substitution in almost all cases. The risk that PEF with B = 0.6 will give incorrect incentives for incineration or recycling is very small.

However, this accuracy is a bit of a coincidence. Factor A in CFF varies between materials. PEF results for polymers with B = 0.6 will provide accurate indication regarding the waste management of most polymers (A = 0.5); however, it might give incorrect incentives for incineration of textiles (A = 0.8), and incorrect incentives for recycling of many paper grades (A = 0.2). In addition, Factor B as estimated with our method will vary over time and between locations, depending on to what extent the recovered energy contributes to the expected profitability of the incinerator.

If Factor B is defined to be equal to A, PEF results would almost always rank incineration and recycling accurately. However, we have found no scientific basis for arguing that Factor B should vary between materials, which Factor A does.

Our approach for estimating B is based on the drivers for investments in waste incineration plants. These are only relevant in an LCA aiming to assess the long-term consequences of sending waste to incineration. In the short term the incineration capacity is fixed and hence cannot be affected by a change in the flow of plastic waste.

Our approach accounts for the fact that incineration with energy recovery does not increase simply because a waste flow is sent to the incinerator. This is in parallel to Factor A, which accounts for the fact that recycling does not increase simply because a waste flow is sent to recycling. In this sense, our approach makes the CFF more consistent and balanced. Further studies are required to investigate if the approach is valid also for other energy-recovery processes, such as combustion in cement kilns.

Even if CFF becomes more balanced, it still has the limitation of accounting for consequences of actions in a single product life cycle at a time. It still does not assess in combinations the concerted actions in the life cycle generating waste and the life cycle utilizing the recycled material or recovered energy.

5.3. The European perspective

Scenarios with a European perspective are particularly relevant in the short-term perspective, because the waste incinerator capacity is typically fully utilized, and incinerator capacity is in the short term unaffected by a change in the incinerated flow of plastic waste. Instead, the flow of plastic waste primarily affects the import of waste from other European countries.

The scenarios European incineration and European landfill illustrate the uncertainty in the climate impact of a change in waste imports (Fig. 7d). The large uncertainty means that the difference in climate benefit between recycling and incineration is no longer significant for renewable LDPE.

Just like Factor B above, the expanded systems perspective accounts for the fact that not only recycling but also energy recovery depends on more than the flow of waste from the life cycle investigated. However, the large uncertainty in the European impacts means the PEF results give little guidance on whether a product with renewable LDPE has a lower climate impact if it is incinerated or recycled. In this sense, it removes the incentives for recycling as well as energy recovery.

5.4. Choosing between or combining the two solutions

In the choice between the two solutions, it is worth noting that Factor B is relevant when assessing long-term effects of plastic waste incineration while the waste imports are most clearly affected in the short term. An LCA, just like other environmental assessments, aims at increased long-term sustainability. In this perspective long-term effects, such as investment in new incinerator capacity, are likely to be more important than short-term fluctuations in waste flows and their treatment. This indicates that it is more relevant to estimate Factor B than to estimate the impacts of a change in waste imports.

However, since B > 0, a change in the flow of plastic waste to incinerators will not fully be met by an increase in incinerator capacity. Part of the long-term effects will also include crowding out of imported waste. Comprehensive modelling of the foreseeable long-term consequences of an increased plastic waste flow will require a combined approach: an estimate of Factor B to decide to what extent the increase is met by incinerator investments, and scenarios to model the European impacts of the change in the waste imports.

5.5. Assessing the concerted action

The net environmental benefit of recycling is the joint result of at least two actions: 1. sending material to recycling, and 2. using recycled material instead of virgin material. The net environmental benefit of energy recovery is also the joint result of at least two actions: sending waste to energy recovery and using the energy recovered from waste instead of energy from other sources. The focus on individual life cycles is apparently a limitation with LCA in these cases. None of the solutions discussed here solves this problem completely. To guide coordinated or concerted action, the actions should ideally be assessed in combination (cf. Andreasi Bassi et al., 2021).

Many situations require the environmental impacts to be estimated for a specific product or a specific action. In some cases, however, the LCA results can be jointly calculated and then presented with, for example, the following introduction:

> "When the material is sent to recycling, you will, together with the recycler and the actor using the recycled material, jointly achieve this net environmental benefit: …"

When such statements can be made, the allocation problems of recycling and waste incineration are avoided. There is no longer a need to subjectively choose a method to divide the environmental benefits of recycling or energy recovery between the life cycle generating the waste and the life cycle where the recovered material or energy is used. This solution is in line with the recommendation in the old SETAC "Code of Practice" to assess life cycles with recycling by studying the inputs and outputs from the total linked system (SETAC 1993, p.21).

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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References

Andreasi Bassi, S., Tonini, D., Ekvall, T., Astrup, T.F., 2021. A life cycle assessment framework for large-scale changes in material circularity. Waste Management 135 (this issue), 360–371.

BSI, 2011. The Guide to PAS 2050:2011 - How to carbon footprint your products, identify hotspots and reduce emissions in your supply chain. British Standards Institution, London, UK.

Bisaillon, M., Sahlin, J., Ekvall, T., Sundberg, J., 2005. Driving forces for import of waste for energy recovery in Sweden. Waste Manag. & Res. 23 (1), 3–12. https://doi.org/10.1177/0734242X05051460.

WRI & WBCSD, 2011. Product Life Cycle Accounting and Reporting Standard. World Resources Institute and World Business Council for Sustainable Development, USA.

EC, 2018a. Product Environmental Footprint Category Rules Guidance: Version 6.3 – May 2018. European Commission. url: http://ec.europa.eu/environment/eussd/smgp/pdf/PEFCR_guidance_v6.3.pdf.

EC, 2018b. Results and deliverables of the Environmental Footprint pilot phase. Guidance documents: Annex C. European Commission url: https://ec.europa.eu/environment/eussd/smgp/PEFCR_OEFSR_en.htm.

Ekvall, T., 2000. A market-based approach to allocation at open-loop recycling. Resour. Conserv. Recycl. 29 (1–2), 93–111. https://doi.org/10.1016/S0921-3449(99)00057-9.

EI, 2021. Nu finns information om residualmix för 2019. The Swedish Energy Market Inspectorate. url: https://www.ei.se/sv/nyhetsrum/nyheter/nyheter-2020/nu-finns-information-om-residualmix-for-2019/. (in Swedish).

Ekvall, T., Björklund, A., Sandin, G., Jelse, K., 2020. Modeling recycling in life cycle assessment. Report C551. IVL Swedish Environmental Research Institute, Stockholm, Sweden.

Ekvall, T., Gottfridsson, M., Nilsson, J., Nellström, M., Rydberg, M., Rydberg, T., 2021. Incentives for recycling and incineration in LCA: Polymers in Product Environmental Footprints. Report 2021:02. Swedish Life Cycle Center, Gothenburg, Sweden.

Fråne, A., Youhanan, L., Ekvall, T., Jensen, C., 2016. Avfallsimport och materialåtervinning. Report B2266. IVL Swedish Environmental Research Institute. (in Swedish), Stockholm, Sweden.

Fuji, H., Nagaiwa, T., Kusuno, H., Malm, S., 2005. How to quantify the environmental profile of stainless steel. Paper presented at the SETAC North America 26th Annual Meeting, Baltimore, USA, November 13-17 2005. url: http://www.worldstainless.org/Files/issf/non-image-files/PDF/Howtoquantifytheenvironmentalprofileofstainlessste.pdf.

Gode, J., Fredén, J., Adolfsson, I., Ekvall, T., 2013. Värdering av fjärrvärmens resurseffektivitet och miljöpåverkan – Metodfrågor. Report 2013:3. Svensk Fjärrvärme, Stockholm Sweden. (in Swedish).

Hagberg, M., Gode, J., Lätt, A., Ekvall, T., Adolfsson, I., Martinsson, F., 2017. Miljövärdering av energilösningar i byggnader – Etapp 2. Fjärrsynsrapport 2017: 409. Energiforsk, Stockholm, Sweden. (in Swedish).

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M.D. M., Hollander, A., Zijp, M., van Zelm, R., 2017. ReCiPe 2016 v1.1 – A harmonized life cycle impact assessment method at midpoint and endpoint level – Report I: Characterization. RIVM Report 2016-0104a. National Institute for Public Health and the Environment, Bilthoven, The Netherlands.

IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, USA.

ISO, 2012. Environmental management – Life cycle assessment – Illustrative examples on how to apply ISO 14044 to goal and scope definition and inventory analysis (ISO. In: TR, 14049. International Organization for Standardization, Geneva, Switzerland, p. 2012.

ISO, 2018a. Greenhouse gases — Carbon footprint of products — Requirements and guidelines for quantification (ISO 14067:2018). International Organization for Standardization, Geneva, Switzerland.

ISO, 2018b. Life cycle inventory calculation methodology for steel products (ISO 20915: 18, IDT). International Organization for Standardization, Geneva, Switzerland.

Lassesson, H., Gottfridsson, M., Nellström, M., Rydberg, T., Josefsson, L., Mattsson, C., 2021. Kemisk återvinning av plast – teknik, flöden och miljöaspekter. Report 6990. Swedish Environmental Protection Agency, Stockholm, Sweden. (in Swedish).

Lindfors, L.-G., Christiansen, K., Hoffman, L., Virtanen, Y., Juntilla, V., Hanssen, O.-J., Rønning, A., Ekvall, T., Finnveden, G., 1995. Nordic Guidelines on Life-Cycle Assessment. Nord 1995:20. Nordic Council of Ministers, Copenhagen, Denmark.

JRC, 2010. ILCD Handbook - General guide for Life Cycle Assessment - Detailed Guidance. European Commission Joint Research Centre, Ispra, Italy.

Moora, H., Stenmarck, Å., Sundqvist, J.-O., 2006. Use of life cycle assessment as decision-support tool in waste management – Optimal waste management scenarios for the Baltic states. Env. Eng. & Manag. J. 5 (3), 445–455.

Phyllis, 2020. LDPE (low density PE) (#775) at Phyllis2. url: https://phyllis.nl/Browse/Standard/ECN-Phyllis#ldpe.

Profu, 2013. Tio perspektiv på framtida avfallsbehandling. Rapport från forskningsprojektet "Perspektiv på framtida avfallsbehandling. PROFU, Mölndal, Sweden. (in Swedish).

Schrijvers, D.L., Loubet, P., Sonnemann, G., 2016. Developing a systematic framework for consistent allocation in LCA. Int. J. LCA 21 (7), 976–993. https://doi.org/10.1007/s11367-016-1063-3.

SETAC, 1993. Guidelines for Life-Cycle Assessment: A 'Code of Practice'. Society of Environmental Toxicology and Chemistry, Brussels/Pensacola, Belgium/USA.

Renova, 2020. Annual Report 2019. Gothenburg, Sweden: Renova Miljö AB. (in Swedish).

Waste Sweden, 2012a. Assessment of increased trade of combustible waste in the European Union. Report F2012:4. Waste Sweden, Malmö, Sweden.

Waste Sweden, 2012b. Kapacitetsutredning 2011. Tillgång och efterfrågan på avfallsbehandling till år 2020. Report F2012:3. Waste Sweden, Malmö, Sweden. (in Swedish).

Waste Sweden, 2014. GUIDE #12 Rekommendation avseende miljövärdering av avfallsförbränning med energiåtervinning. Waste Sweden, Malmö, Sweden. (in Swedish).

Waste Sweden, 2019. Svensk Avfallshantering 2018. Waste Sweden, Malmö, Sweden (in Swedish).

Weidema, B., 2001. Avoiding co-product allocation in life cycle assessment. J. Ind. Ecol. 4 (3), 11–33. https://doi.org/10.1162/108819800300106366.

Worldsteel, 2017. Life Cycle Inventory Methodology Report. World Steel Association, Brussels, Belgium.